Impact of Swine Waste Application on Ground and Stream Water Quality in an Eastern Coastal Plain Watershed

K. C. Stone, P. G. Hunt, F. J. Humenik, M. H. Johnson

Abstract. *Nonpoint source pollution from agriculture has been a major concern, particularly where intensive agricultural operations exist near environmentally sensitive waters. To address these concerns, a water quality project was initiated in Duplin County, North Carolina, in the 2044-ha Herrings Marsh Run watershed. A swine farm within this monitored watershed expanded its operation from 3,300 to more than 14,000 animals. Groundwater nitrate-N increased significantly in three of the seven wells located adjacent to the spray field and in the adjoining riparian zone. Stream nitrate-N concentrations have increased after the expansion of the swine operation in the colder months, but concentrations have remained approximately the same during the warmer months. Stream ammonia-N mean concentrations after expansion have increased as well as the frequency and magnitude of ammonia-N concentration spikes. Ortho-phosphate concentrations in the stream water have been relatively consistent over the study period. The riparian zone is reducing the impact of spray field groundwater nitrate concentrations and ammonia loadings in an adjacent stream.* ***Keywords.*** *Water quality, Nitrate, Nonpoint source pollution, Swine waste.*

Nonpoint source contamination of ground and stream water by agricultural chemicals is a major public concern throughout the USA as well as in the eastern Coastal Plain. Nitrate contamination in groundwater is a particular concern for both health and environmental quality. Groundwater is the major source of drinking water for more than 90% of rural households and 75% of cities in the USA (Goodrich et al., 1991). In the Coastal Plain, ground and stream waters are very closely linked. Excess nutrients in the groundwater can lead to stream water contamination which can lead to excessive plant and algae growth that can exhaust oxygen supplies in water and result in fish kills and loss of desirable aquatic vegetation.

When nutrients are applied in excess of the crop's ability to use in a harvestable product, they may be lost to the environment. Many fields in the eastern Coastal Plain are multi-cropped, which requires several applications of various pesticides and nutrients. Nutrient leaching to groundwater is a potential problem because of high rainfall, sandy textures, and low soil organic matter levels. Nutrients can also reach streams from overland flow or from lateral movement of shallow groundwater. Nutrient leaching and runoff are a concern because of the large amounts of swine and poultry waste being produced in the eastern Coastal Plain. Adoption of improved management practices can help reduce the potential of these chemicals being lost to the environment.

Since 1988, the swine population in North Carolina has risen from approximately two million to more than eight million (USDA-NASS, 1995). Operation size is also a concern relative to water quality with 86% of the swine population produced on farms with greater than 2,000 head (USDA-NASS, 1995). This rapid expansion of the swine industry and use of industrial methods for production have led to environmental concerns. In addition to swine, poultry is extensively produced in the eastern Coastal Plain. Approximately 80 million turkeys and chickens are produced annually in North Carolina alone (USDA-NASS, 1995). Production of waste from these sites is often greater than nutrient demand by local crops. Barker and Zublena (1995) reported that several counties in North Carolina produced more nitrogen in plant-available nutrients from animal manure than needed by non-legume agronomic and forage crops. Together, intensive crop and animal production pose a great contamination potential if adequate nutrient management practices are not implemented. Natural landscape characteristics of eastern Coastal Plain watersheds, such as large wooded riparian zones and soils with high organic matter, typically have helped mitigate elevated nutrient levels from reaching streams and shallow groundwater (Gilliam, 1991). However, with the large influx of animal production and limited land for waste application, these natural features can become overloaded and their effectiveness negated.

To address these environmental concerns, a Water Quality Demonstration Project involving federal, state, and local agencies; private industry; and local landowners was initiated in 1990 on a watershed in the Cape Fear River Basin in Duplin County, N.C. (Stone et al., 1995). The demonstration watershed, Herrings Marsh Run (HMR), has

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many characteristics typical of an intensive agricultural area in the eastern Coastal Plain of the USA (Hubbard and Sheridan, 1989). Duplin County has the highest agricultural revenue from livestock of any county in North Carolina and is second in total revenue to neighboring Sampson County (North Carolina Dept. of Agriculture, 1996).

On the HMR watershed, one of the farms that was monitored to assess the impact of agricultural operation on the environment was a swine farm (fig. 1). The swine farm was typical of similar farms throughout the county and region. Originally, the swine farm had approximately 3,300 head; however, near the middle of the study period, the swine operation was expanded. The expansion was from one production facility to four facilities with a swine population of greater than 14,000 head. The objective of the study was to determine the ground and stream water quality changes resulting from a large swine operation expansion in the North Carolina Coastal Plain.

METHODS

WATERSHED AND SITE DESCRIPTION

The Herrings Marsh Run (HMR) watershed is located in the Coastal Plains physiographic region of Duplin County, North Carolina. The HMR watershed contains 2,044 ha and is centered at approximately lat. 35°06′ North and long. 77°56′ West. In the southwest section of the Herrings Marsh Run Watershed, a 32-ha swine waste water spray field, adjacent 16-ha riparian zone, and stream were monitored to determine nutrient loading of the ground and stream water (fig. 1). The spray field was originally in row crop rotation and later converted to a coastal bermuda pasture. The riparian zone comprised approximately 1.2 to 2 ha of hardwood swamp forest in a band approximately 30 m wide along the creek and 15 ha of pine-mixed hardwood forest throughout the uplands.

In 1995, the swine operation was expanded by three additional finishing houses, one in the spray field near well 3 and one close to wells 4 and 5 (fig. 1). No other swine operations were located in this sub-watershed. In 1995, swine population increased from 3,300 head to more than 14,000 head. Plant available nitrogen increased from approximately 3.5 Mg/year to 15 Mg/year. The spray field was expanded to include more than 48 ha and was in coastal bermuda grass that was over-seeded in winter. The coastal bermuda was harvested for hay, and some areas were rotationally grazed. Potential nitrogen application rates to the coastal bermuda spray field increased from approximately 100 kg/ha to 300 kg/ha.

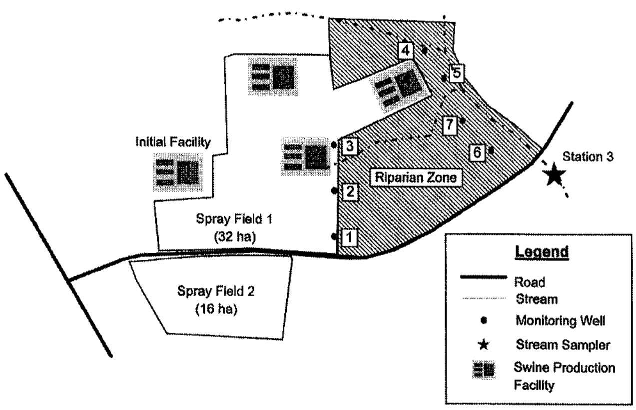

Figure 1–Location of wells and stream sampling station for spray field and riparian zone study area.

GROUNDWATER

Three groundwater monitoring wells were installed at the edge of a swine waste water spray field at a depth of approximately 7.6 m and were used to monitor nutrient changes in 1993. Four monitoring wells (4-7) installed at a depth of approximately 1.5 m, placed in transects parallel to the stream in the riparian zone in 1994 (fig. 1), were monitored to measure constituent changes in groundwater quality through the riparian zone.

Local topography was used as a guide for determining groundwater gradients, and input from landowners and farmers was used to determine groundwater monitoring well placement with the goal of minimizing interference with normal farming activities. The wells were installed using a SIMCO 2800 trailer-mounted drill rig equipped with 108-mm inside-diameter hollow stem augers. The well casings and screen were 50-mm threaded schedule 40 PVC, and well screens were 1.5 m long. On shallow wells, a 0.75-m screen was used. Well bottoms were placed on an impermeable layer or to a depth of 7.6 m if the impermeable layer could not be located above that depth. Water table depths in the watershed were generally 1.5 to 3 m below the soil surface. Monitoring wells were constructed according to North Carolina Department of Environmental Management regulations. A filter pack of coarse sand was placed around the well screens. An annular seal of bentonite was placed above the filter sand. Concrete grout was then placed above the bentonite to the soil surface to prevent contamination from the surface. Locking well covers were installed to prevent unauthorized access. WaTerra foot valves (model D-25) and high density polyethylene tubing were installed in each well to provide dedicated samplers.

Collection of shallow groundwater from monitoring wells began in 1993. Before each sample was collected, the static well water depth was measured, and one to three well volumes were purged. A glass sample collection bottle was rinsed with the well water, filled with a sample, packed in ice, and transported to the laboratory. Wells were sampled monthly.

STREAM WATER

Stream water quality samples were collected by an automated water sampler immediately downstream from the riparian zone (fig. 1). Lack of a distinct stream channel upstream of this feature rendered collection of meaningful data upstream of this riparian zone impossible. Samples were collected using automated samplers (ISCO and American Sigma).

Samples were collected automatically on a timed sequence. During the period 6 September 1990, until 13 October 1993, the sampler collected samples every two hours and composited them into a daily sample. After 14 October 1993, the sampler collected two-hour samples and composited them into a 3.5-day sample. Samples were preserved by acidifying to a pH < 2 with sulfuric acid. Samples were retrieved from the samplers weekly, packed in ice, and transported to the laboratory. They were then stored in refrigerators at 4°C until chemical analyses were performed.

All water samples were transported to the USDA-ARS, Soil, Water, and Plant Research Center in Florence, South Carolina, for analysis. Water samples were analyzed using a TRAACS 800 Auto-Analyzer for nitrate-N, ammonia-N, and ortho-phosphorus using EPA Methods 353.2, 350.1, and 365.1, respectively (U.S. EPA, 1983). EPA-certified quality control samples were routinely analyzed to verify results.

Stream flow measurements were recorded by the U.S. Geological Survey for station 3 downstream of the riparian zone (USGS Station: 0210783230 Herrings Marsh Run near Summerlins Crossroads, N.C.). Nutrient loading rates were calculated by merging nutrient data and stream flow data.

Statistical Analysis

Statistical analyses on the collected stream and groundwater samples were performed using the SAS system (SAS, 1990). The nutrient data were not normally distributed and were log-transformed. Means of the log-transformed data were calculated and then back-transformed resulting in geometric means. A regression analysis of stream and groundwater data was performed to determine if any significant trends in nutrient concentrations existed during the study period. A t-test was performed on the data to determine statistical differences in nutrient concentrations in water samples collected before and after expansion of the swine operation. The data were then sub-divided into warm and cold seasons. Warm seasons data contained data for the months April-November, while cold season data was for December-March. A t-test was then used to determine statistical differences in nutrient concentrations and mass fluxes between warm and cold season.

Results and Discussion

Groundwater

Groundwater nitrate-N concentrations increased in four (three significantly) of the seven monitoring wells from 1993 to 1997 (table 1). Two of these four wells nearly doubled in nitrate-N concentration, and by 1997 their nitrate-N concentrations exceed the drinking water standard of 10 mg/L. When groundwater nitrate-N data were regressed against time, wells 1, 3, and 4 had slopes significantly ($P \leq 0.01$) different from zero and r^2 values greater than 0.60 (table 2 and figs. 2 and 4). Well 2 had a significant slope at the $P \leq 0.06$ level but had a low r^2 value of 0.09 (table 2 and fig. 3). The other three wells (5,

Table 1. Mean yearly groundwater nitrate-N concentrations for monitoring wells

	Nitrate-N (mg/L)						
	Well Number						
Year	1	2	3	4	5	6	7
1993	5.8	7.6	6.3				
1994	6.1	7.7	6.4	1.0	7.2	0.09	3.4
1995	8.4	7.9	8.6	2.8	7.7	0.06	3.5
1996	10.0	8.2	10.6	5.5	6.7	0.56	3.3
1997	11.1	8.7	12.1		7.8	0.15	2.7
$LSD_{0.05}$	1.2	1.2	1.2	1.6	1.1	1.8	1.4

Table 2. Groundwater monitoring well regression analysis results

Well	A*	Prob > |T|	B*	Prob > |T|	r^2
1	3.016	0.0001	5.606e-04	0.0001	0.634
2	6.852	0.0001	9.075e-05	0.0524	0.083
3	3.492	0.0001	5.236e-04	0.0001	0.608
4	0.017	0.0001	2.803e-03	0.0001	0.641
5	7.783	0.0001	−1.640e-05	0.7704	0.003
6	0.034	0.0001	4.955e-04	0.1832	0.076
7	4.449	0.0002	−1.626e-04	0.3779	0.030

* A and B are regression coefficients for the equation:
$\text{Nitrate-N} = A \times e^{(B*DAY)}$
where DAY is the number of days since 6 September 1990, when monitoring at the site began.

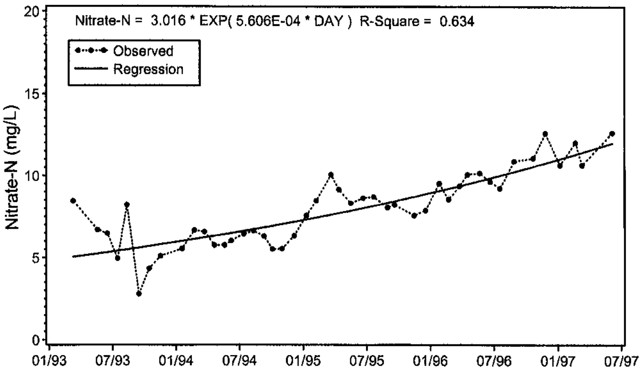

Figure 2–Groundwater nitrate-N concentrations and regression for well 1.

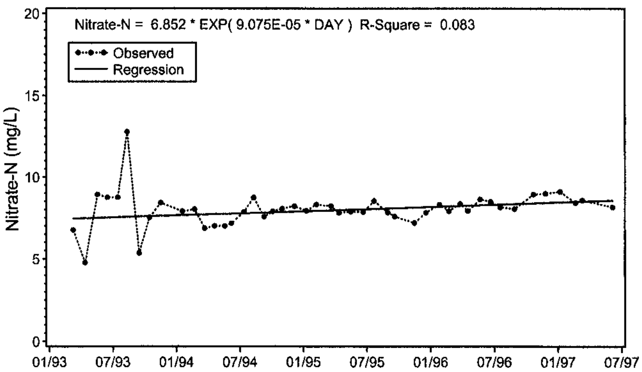

Figure 3–Groundwater nitrate-N concentrations and regression for well 2.

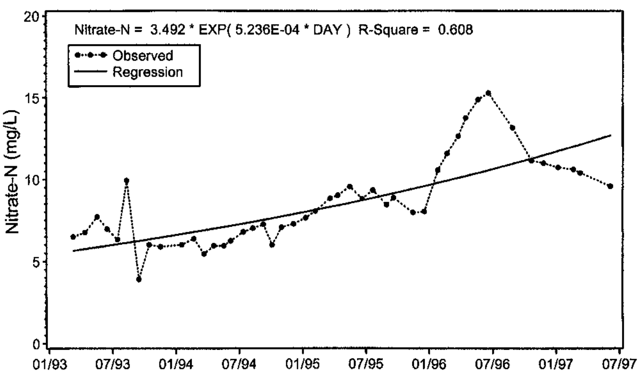

Figure 4–Groundwater nitrate-N concentrations and regression for well 3.

6, 7) had slopes that were not significantly different from zero. Wells 6 and 7 were located in the riparian zone (fig. 1), and they had nitrate-N concentrations of 0.2 and 3.2 mg/L, respectively, throughout the study period.

Wells with a significant increasing trend in groundwater nitrate-N appear to have been influenced by the swine operation even before expansion and the increased land application of swine waste after expansion. Wells 3 and 4 could have also been affected by their proximity to the newly installed facilities and lagoons. Huffman and Westerman (1995) found that four of eleven lagoons studied in the lower Coastal Plain of North Carolina had moderately high seepage rates. They found that older lagoons generally had more losses, but that inadequate or improper lining of lagoons could also allow seepage. However, well 1 was not located in close proximity to the new lagoons and had an increasing slope similar to that of wells 3 and 4. This would suggest that the problem was not seepage from the lagoon.

The increase in groundwater nitrate-N was more likely due to the wastewater application timing, rate, or distribution. As previously mentioned, the operation was expanded in 1995 from 3,300 head to over 14,000 head and from one facility to four facilities. Plant available nitrogen increased from approximately 3.5 Mg/year to 15 Mg/year. Estimated land area to adequately use this amount of waste was approximately 50 ha with continuous cover and winter over seeding. Nitrogen application rate estimates of approximately 300 kg/ha/yr may have been an over-estimate of the coastal bermuda grasses ability to take up nitrogen at the high application rates.

Stream Water

Stream nutrient levels at station 3 generally have been superior to other stations and subwatersheds in the HMR watershed (Stone et al., 1995), but nutrient levels have been above those found in nonimpacted watersheds (Duda, 1984). Mean nitrate-N at station 3 was 1.12 mg/L for the study period. Initially, mean nitrate-N was 0.95 mg/L in 1990, but it has since increased to 1.33 and 1.85 mg/L in 1996 and 1997, respectively (table 3). A similar increase occurred for ammonia-N. In 1990 and 1991, mean ammonia-N concentration was 0.05 mg/L, and it has increased to 0.32 mg/L in 1997. Ortho-P concentrations have not changed appreciably during the study period with a mean of 0.068 mg/L.

A regression analysis for the nutrients was conducted on the stream water to determine if any trends existed during the study period (table 4). Stream nitrate-N concentrations have shown a slight increase over the study period (fig. 5), but the simple linear regression slope was not significantly different from zero. This may be related to seasonal cycling of nitrate-N which will be discussed in the following section.

Table 3. Mean yearly stream nutrient concentrations and mass fluxes for station 3 on the Herrings Marsh Run Watershed

Year	Nitrate-N (mg/L)	Ammonia-N (mg/L)	Ortho-P (mg/L)	Mass Nitrate-N (kg/day)	Mass Ammonia-N (kg/day)	Mass Ortho-P (kg/day)	Flow (m^3/s)
1990	0.95	0.05	0.05				
1991	1.15	0.07	0.06	3.73	0.21	0.21	0.037
1992	1.00	0.17	0.10	2.36	0.35	0.22	0.026
1993	1.61	0.19	0.09	4.47	0.49	0.23	0.018
1994	0.74	0.11	0.05	0.88	0.10	0.04	0.015
1995	1.15	0.23	0.06	2.49	0.51	0.18	0.026
1996	1.33	0.22	0.04	3.63	0.61	0.10	0.033
1997	1.85	0.32	0.05	14.91	1.87	0.16	0.080
$LSD_{0.05}$	1.17	1.31	1.33	1.81	1.73	1.97	1.300

Table 4. Stream nutrient regression analysis results

Nutrient	A*	Prob > |T|	B*	Prob > |T|	r^2
Nitrate-N	0.984	0.9185	6.870e-05	0.5270	0.006
Ammonia-N	0.085	0.0001	4.644e-04	0.0003	0.177
Ortho-P	0.097	0.0001	−3.618e-04	0.0097	0.096
Mass Nitrate-N	2.149	0.0647	6.432e-05	0.0821	0.001
Mass Ammonia-N	0.179	0.0001	4.967e-04	0.0169	0.082
Mass Ortho-P	0.221	0.0001	−3.828e-04	0.0571	0.053

* A and B are regression coefficients for the equation: Nutrient = $A \times e^{(B*DAY)}$ where DAY is the number of days since 6 September 1990, when monitoring at the site began.

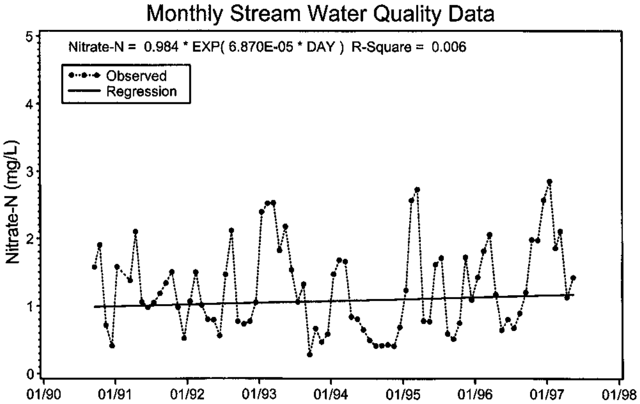

Figure 5–Monthly stream water nitrate-N concentrations and regression.

Regression analyses of the nitrate-N mass flux loading rates follow the same trend as the nitrate-N concentrations. The nitrate-N mass flux slope significantly ($P \leq 0.08$) increased during the study, but it had a low r^2 value. Overall, nitrate-N concentrations and mass loading leaving the watershed appear to have remained fairly constant over the study period probably because of very high denitrification rates in the riparian zone.

Stream ammonia-N concentrations have had an increasing trend over the study period. In 1990, mean ammonia-N concentration was 0.05 mg/L, and it has increased during the study (table 3), with 1995-1997 having the highest ammonia-N concentrations (0.22-0.32 mg/L).

A regression analyses of the stream ammonia-N concentrations show that the ammonia-N had a significant increasing slope during the study period. Figure 6 also shows that the ammonia-N concentrations have increased more during the last few years. This increase in ammonia-N concentrations is probably related to the increasing frequency and magnitude of ammonia-N spikes observed (fig. 7). Before the summer of 1995, the frequency of ammonia-N peaks at the natural riparian zone was very low, and few of the peaks exceeded 1.0 mg/L (fig. 7). However, after 1995, the peaks began to exceed 2 mg/L, and their frequency has increased. Also, a similar

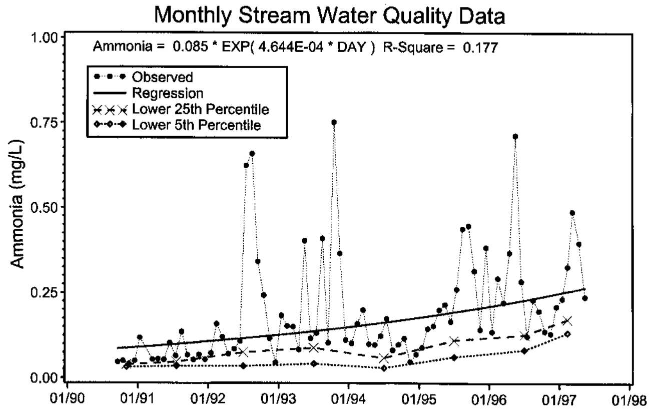

Figure 6–Monthly stream water ammonia-N concentrations, regression, 25th and 5th percentiles.

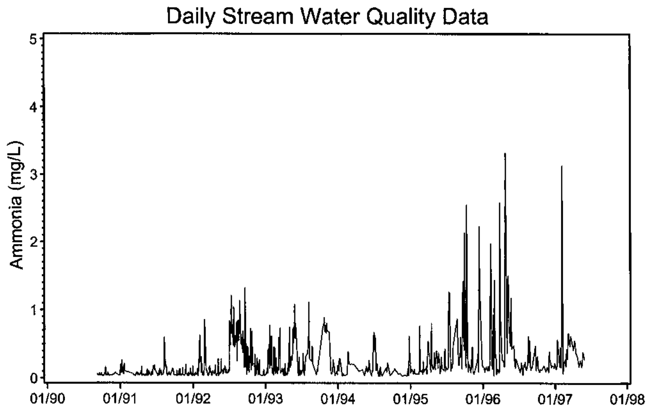

Figure 7–Daily stream water ammonia-N concentrations.

increasing trend in ammonia-N concentrations was observed in the yearly lower 5th and 25th percentile concentrations from 1995-1997 (fig. 6). These results seem to indicate that the stream is getting an ammonia-N source that did not exist before 1994. This source is most likely the expanded swine operation immediately upstream from the stream sampler. Ammonia-N is an oxidatively reduced compound that is oxidized in a relatively short time in an aerobic soil profile. Thus, its presence in the stream water indicates recent overland flow contamination that is more event-controlled.

Similar to the ammonia-N concentrations, ammonia-N mass fluxes have increased during the study. A regression analysis of the ammonia-N mass fluxes shows a significantly increasing slope ($P \leq 0.02$, table 4).

The stream ortho-P concentrations have remained steady over the study period except for 1992-1993. Ortho-P concentration in 1990 was 0.05 mg/L and in 1997 was approximately the same. Table 3 shows that 1992 and 1993 had higher concentrations than the other years.

Regression analysis of the ortho-P concentrations showed a significantly decreasing slope ($P = 0.01$); however, the slope coefficient was very low and figure 8 shows that the ortho-P concentrations have remained fairly constant and may have decreased slightly over the study period (table 4, fig. 8). The lack of elevated ortho-P in the stream indicates that applied phosphorus is being removed

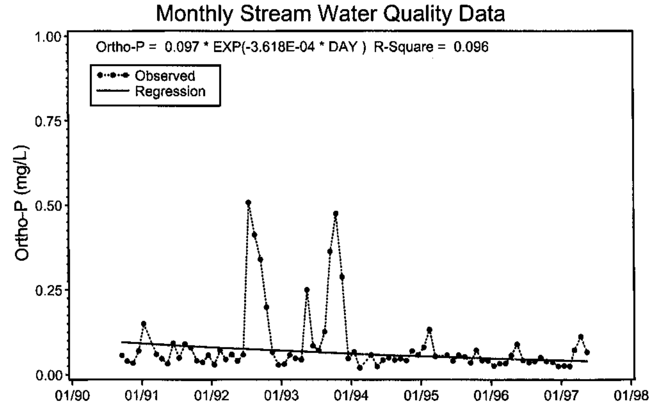

Figure 8–Monthly stream water ortho-P concentrations and regression.

by the ecosystem in the riparian zone and/or spray field prior to runoff reaching the stream.

Similar to the ortho-P concentration data, ortho-P mass flux has decreased slightly over the study period (table 3). Regression analysis of the mass ortho-P fluxes (table 4) indicated a significant decreasing trend ($P = 0.06$); however, like the ortho-P concentrations data, the slope coefficient was small.

Seasonal and Before/After Expansion Analysis

Seasonal nitrate-N concentrations were significantly different both before and after expansion of the swine operation (table 5). Warm season nitrate-N concentrations were not significantly different from each other either before or after expansion. This may be because of the high denitrification capacity of the stream and riparian zone during the warmer months. However, cold season nitrate-N concentrations were significantly higher than warm season concentrations and also were significantly different between the time periods, with means of 1.22 and 1.89 mg/L before and after expansion, respectively. This would indicate that during the warmer months, the riparian zone was able to buffer the increased loading from the swine expansion while during the colder months, which

Table 5. Means of nutrient concentrations and mass fluxes for before and after swine production operation expansion and for cold and warm seasonal analysis

Season	Nitrate-N	Ammonia-N	Ortho-P	Mass Nitrate	Mass Ammonia-N	Mass Ortho-P	Flow
	------------(mg/L)------------			-----------(kg/day)-----------			(m^3/s)
1990-1994 Before Expansion of Swine Production							
Cold*	1.22 a†	0.10 a†	0.05 a†	8.19 a	0.67 a†	0.28 a†	0.066 a
Warm	1.04 b	0.13 a‡	0.08 b‡	1.60 b	0.22 b‡	0.15 b‡	0.014 b‡
1995-1997 After Expansion of Swine Production							
Cold	1.89 c†	0.22 c†	0.04 c†	9.94 c	1.17 c†	0.21 c†	0.060 c
Warm	1.02 d	0.24 c‡	0.05 c‡	1.63 d	0.40 d‡	0.09 d‡	0.021 d‡

Cold season (December-March), Warm season (April-November).

* (a, b, c, e) Cold and warm season means followed by the same letter are not significantly different at the 0.05 level using a t-test for the same time period.

† Cold season means with an † were significantly different at the 0.05 level using the t-test.

‡ Warm season means followed with an ‡ were significantly different at the 0.05 level using the t-test.

have less plant growth and microbial denitrification, the riparian zone could not assimilate the increased loading.

Warm season nitrate-N mass fluxes were not significantly different from each other with means of approximately 1.6 kg/day. This pattern was similar to the nitrate-N concentrations' findings. Additionally, even during the cold season, nitrate-N mass fluxes were not significantly different between before and after expansion with means of 8.19 and 9.94 kg/day, respectively. However, the cold season nitrate-N mass fluxes were significantly higher than the warm seasons fluxes both before and after expansion. These results from nitrate-N concentration and mass fluxes indicate that nitrate-N in the stream water has not been affected appreciably from the swine operation expansion.

Ammonia-N concentrations were not affected by the seasons either before or after expansion. However, ammonia-N concentrations were significantly higher after the swine operation was expanded. Before expansion of the operation, ammonia-N concentrations averaged approximately 0.11 mg/L. After expansion of the operation, ammonia-N concentrations more than doubled to approximately 0.23 mg/L. These data indicate that the stream is getting a continuous source of ammonia-N that did not exist prior to the expansion of the operation. Similar to the ammonia-N concentration, ammonia-N mass transport in the stream has also increased significantly after expansion of the swine operation.

Ortho-P concentrations have decreased after expansion of the swine operation for both the warm and cold seasons as have the ortho-P mass fluxes. After expansion of the operation, ortho-P concentrations were not significantly different for the cold and warm seasons, and the ortho-P mass flux was significantly higher during the colder months primarily due the increased stream flows.

Conclusions

1. Groundwater nitrate-N has been impacted by the application of swine waste water applications. Three wells had a significant increase in nitrate-N concentrations. Two wells at the field edge now exceed the safe drinking water level of 10 mg/L nitrate-N. However, only one well in the riparian zone was affected, and nitrate-N in this well remains below the safe drinking water level.
2. Stream nitrate-N concentrations have increased after the expansion of the swine operation during the colder months of the year, but they have remained approximately the same during the warmer months.
3. Stream ammonia-N concentrations have significantly increased after expansion of the swine operation. The frequency and magnitude of ammonia-N concentration spikes have also increased after expansion.
4. Ortho-P concentrations in the stream water have been relatively consistent over the study period.
5. The riparian zone appears to be providing a significant buffer to both ground and stream water during the warmer months.

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